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Chapter 3:
Threats

The continued survival of the southwestern willow flycatcher (*Empidonax traillii extimus*) is threatened indirectly by the loss, modification, and fragmentation of riparian habitat, and directly by factors that impact the survival and reproductive success of flycatchers. Because the impact of habitat loss on small populations can be particularly severe, we first discuss some of the population-level effects that may be influencing flycatcher population dynamics. We then review some contemporary cases of habitat loss and discuss other factors potentially impacting the flycatcher. The effects of brown-headed cowbird (*Molothrus ater*) parasitism on the southwestern willow flycatcher are discussed in detail in Chapter 8. For additional information on site-specific threats to the southwestern willow flycatcher readers should consult Greenwald (1998).

Effects of Habitat Loss and Fragmentation

Habitat loss and habitat fragmentation are interrelated processes that affect patterns of species' abundance and distribution at local and regional scales (Pulliam and Dunning 1994). Habitat loss is the reduction of the total amount of a particular habitat type in a landscape. Fragmentation is the apportionment of the remaining habitat into smaller, more isolated patches (Wilcove et al. 1986, Saunders et al. 1991). Habitat loss is often manifested as the conversion of one habitat type to another (e.g., conversion of a forested floodplain to agricultural fields). By reducing the amount of space that can be occupied, habitat loss reduces the total number of individuals that can occur at a particular location or throughout a region.

Riparian habitats in the Southwest are naturally rare and patchy, subject to periodic disturbance and occurring as widely-separated ribbons of woodland and forest within a primarily arid landscape. In Arizona, for example, riparian habitat comprises less than 0.5% of the landscape (Strong and Bock 1990). The actual extent of habitat suitable for the southwestern willow flycatcher is much less. Wide-ranging or highly mobile species that rely on naturally patchy and ever-changing habitats, such as the flycatcher, persist at regional scales as metapopulations, or local breeding groups that are linked together and maintained over time by immigration/emigration and dispersal (Hanski and Gilpin 1991, Pulliam and Dunning 1994). Persistence of local breeding groups is a function of the group's size (numbers of individuals), productivity, survivorship, and the ability of individuals to disperse from one breeding location to another (Harrison 1991). By isolating habitat patches, fragmentation reduces the chance of an individual successfully finding suitable habitat. Searching for increasingly isolated patches leaves individuals vulnerable to mortality from competition, starvation, or predation and can result in delayed or lost of breeding opportunities. Weins (1996) noted that habitat loss is probably the most important factor governing population dynamics when the landscape still contains a high proportion of suitable habitat, but "at a certain threshold of habitat loss, patch isolation may quickly come to dominate population dynamics."

Effects of Small Population Size

Demographic Effects

The overall southwestern willow flycatcher population is small with an estimated 549 territories rangewide (see Chapter 3). Moreover, these territories are distributed among a number of very small breeding groups and only a handful of relatively large breeding groups. The small size of flycatcher populations leaves them vulnerable to local extirpation through environmental stochasticity (e.g., floods, fire, severe weather events, disease), and demographic stochasticity (e.g., shifts in birth/death rates and sex ratios). Even moderate variation in stochastic factors that might be sustained by larger populations can reduce a small population below a threshold level from which it cannot recover. This is especially true with short-lived species such as the southwestern willow flycatcher (see Chapter 7).

The persistence of small populations frequently depends on immigration from nearby populations, at least in some years (Stacey and Taper 1992). The small, isolated nature of current southwestern willow flycatcher populations exacerbates the risk of local extirpation by reducing the likelihood of successful immigration among populations. McCarthey et al. (1998) presented data for 36 sites in Arizona where two or more years worth of survey work had been completed between 1993 and 1997. They documented extirpation at ten sites for a loss of 13 territories, and population declines at an additional 15 sites for a loss of 56 territories. Of the 25 sites that were extirpated or that experienced declines, all but four were small sites comprised of ten or fewer territories. Five of the 36 sites had no change in the number of territories and six sites saw increases in the number of territories for a total gain of 38 territories. Overall, at the 36 sites monitored there was a net loss of 18 territories.

Genetic Effects

Small populations tend to be characterized by low levels of within-population genetic variation, and possibly inbreeding. These conditions may lead to reduced survival, reduced fecundity, lowered resistance to parasites and disease, or physiological abnormalities (Allendorf and Leary 1986, Hartl 1988). Low effective population size also threatens small populations. Effective population size is an index of the actual number of individuals breeding and the number of offspring they contribute to the next generation. The effective population size for a species may be much smaller than the censused population size because of uneven sex ratios, uneven breeding success among females, polygyny (e.g., Sedgwick and Knopf 1989), and low population numbers which exacerbate the above factors.

Synthesizing recent empirical and theoretical studies on population genetics, Lande (1995) suggested that the number 500, long held by some in the conservation biology community to represent the minimum effective population size necessary to maintain a viable population of any species, is far too small. Lande contended that effective population sizes should be much larger (in the range of 5000) in order for a species to maintain normal levels of potentially adaptive genetic variance to counteract the effects of random genetic drift. Lande concluded that, because recovery goals for listed species are often not much higher than the actual population size at the time of listing, maintenance of adequate evolutionary potential and long-term genetic viability was doubtful unless populations were recovered to much larger sizes. Based on Lande's hypothesis, and considering the current status of the flycatcher rangewide, the effective population size for the southwestern willow flycatcher may be critically low.

Factors Contributing to Habitat Loss

Water Management

Dams and Reservoirs—Most of the major and many of the smaller Southwestern rivers support one or more dams that have severely altered the distribution, extent, and species composition of riparian habitats (e.g., Colorado River, Gila River, Kern River, Mojave River, Rio Grande, Salt river, San Diego river, Santa Ana River, Sweetwater River, Tijuana River, Verde River). For example, Mearns (1907; in Ohmart et al. 1988) estimated that the lower Colorado River contained more than 161,900 ha of native riparian habitat at the onset of the 20th Century (prior to the construction of any dams). Nearly 100 years later and with the addition of eight dams and diversions along the 660 km of river between Pearce Ferry and the border with Mexico, the U.S. Bureau of Reclamation (1996) estimated the current extent of native riparian habitat at approximately 1,800 ha, or one percent of its former estimated total.

Riparian habitats are modified, reduced, or lost downstream of dams as a result of changes in flood frequency and duration. Hydrological cycles below dams are modified such that minimum flow events occur with greater frequency and longer duration reducing instream flows and lowering watertables. In some cases, sustained high flows have caused loss of riparian stands from prolonged inundation. For example, Hunter et al. (1987a) documented the loss of a 120 ha stand of cottonwood-willow at the confluence of the Bill Williams River and the Colorado River in 1981 after 24 months of continual high flows released from Alamo Lake. Dams also inhibit annual cycles of flood-induced sediment deposition, floodplain hydration and flushing, and seed dispersal necessary for the establishment and maintenance of riparian habitats

Despite these modifications, some southwestern drainages still have the capacity to develop substantial stands of native cottonwood-willow. Several thousand acres of cottonwood and willow developed along the lower Colorado River below Yuma, AZ after the floods of 1993 (U.S. Bureau of Reclamation 1996). However, due to the diversion of Colorado River water upstream at Moreles Dam, those stands declined from desiccation. Restoration of flows to the lower portion of the Colorado River could substantially increase the extent of riparian habitat on that system simply by maintaining vegetation that becomes established after natural flood events.

The filling of reservoirs results in the loss of riparian habitats upstream of dams. For example, the flooding of Glen Canyon resulted in the loss of southwestern willow flycatchers, which Behle and Higgins (1959) considered a common species. Over time, however, some reservoir inflows have developed extensive deltas colonized by some of the largest stands of riparian trees and shrubs currently found in the Southwest, such as at the head of Elephant Butte Reservoir in New Mexico; at the Salt River and Tonto Creek inflows to Roosevelt Lake in Arizona; the Gila River inflow to San Carlos Reservoir in Arizona; the Colorado River inflow to Lake Mead in Arizona; and the inflow of the South Fork Kern River at Lake Isabella in California. In addition, these areas (except San Carlos Reservoir) support or have supported some of the largest southwestern willow flycatcher populations rangewide (Hubbard 1987, Whitfield and Strong 1995, Sferra et al. 1997, McKernan 1997).

However, current water management policies do not support management strategies to protect and maintain these significant riparian stands. As a result, occupied southwestern willow flycatcher habitat has been lost and flycatchers have suffered nest losses or been displaced. For example, inundation at the inflow to Elephant Butte Reservoir in New Mexico during the 1980s resulted in the loss of willow habitat and displacement of at least 10 flycatcher pairs (Hubbard 1987, T. Schrader U.S. Bureau of Reclamation, pers. comm.). Approximately 283 ha of willow habitat were modified (i.e., loss of understory vegetation) due to inundation at the South Fork Wildlife Area at Lake Isabella in 1995. That event resulted in loss of flycatcher nests and subsequent decline in the number breeding flycatchers in the South Fork Wildlife Area (Whitfield and Strong 1995, USFWS 1997a). Approximately 445 ha of occupied Goodding willow (Salix gooddingii) habitat at the inflow to Lake Mead were anticipated to be lost during the 1997 and 1998 growing seasons due to prolonged inundation (USBR 1996, USFWS 1997b). The number of flycatcher territories and nesting attempts at the inflow decreased in 1997 with increasing levels of inundation (see Chapter 3) . And finally, the habitat at Roosevelt Lake, which supports one of Arizona's largest flycatcher populations, is anticipated to be lost when inflows are sufficient to fill the newly-created reservoir conservation space (USFWS 1996). The deltas associated with these and other reservoirs represent some of the most significant management opportunities available to restore a portion of the extensive riparian habitats historically found on these drainages. They also represent significant opportunities to conserve the southwestern willow flycatcher and the suite of riparian-dependent species found in Southwestern riparian systems.

Diversions and Groundwater Pumping—Surface water diversions and groundwater pumping for agriculture, industrial use (e.g., mining), and municipal use are considered major factors in the deterioration of riparian habitats (USFWS 1993, Briggs 1996). Surface diversions and overdraft of groundwater lower watertables and reduce surface flows. The Arizona Game and Fish Department estimated that in Arizona, alone, more than 1448 km miles of formerly perennial stream are no longer perennial. One of the most extensive stands of native riparian habitat in Arizona along the upper San Pedro River is threatened by increased groundwater withdrawal by the nearby city of Sierra Vista (Davis 1995 [in Briggs 1996]). This threat is particularly ironic in light of the fact that ten years of livestock removal from the San Pedro Riparian National Conservation Area has resulted in a dramatic comeback of cottonwood-willow habitat as well as the return of breeding southwestern willow flycatchers (Krueper 1993, McCarthey et al. 1998). Similarly, human population growth in the Verde watershed has raised concerns that central Arizona's most important aquatic and riparian corridor, the Verde River, will not support riverine, riparian, and aquatic resources over the long term (Verde Watershed Association 1998).

The combination of severe drought and upstream diversion for agricultural use was thought to be the cause of southwestern willow flycatcher territory loss or abandonment of at least eight territories along the middle Rio Grande in the vicinity of San Marcial, New Mexico (D. Leal, USFWS, pers. comm., Cooper 1997).

Land-Use Practices

Channelization and Bank Stabilization—Flood control projects generally shorten, straighten, and narrow river channels with the aim of producing unobstructed pathways to convey floodwaters. These projects can severely reduce the extent of alluvial-influenced floodplain by cutting off main channels from side channels and adjacent floodplains and by reducing meander patterns, which slow stream velocity and dampen the effects of flooding (Poff et al. 1997). Channelization alters stream banks, typically elevating them well above groundwater levels and thus preventing the roots of most native riparian shrubs and trees from accessing groundwater. Overbank flooding necessary to deposit sediments, disperse seeds, rehydrate floodplain soils, and flush accumulations of salts, is reduced or precluded. Channel cutting further reduces water tables adjacent to the river, precluding seedling establishment because of the increased depth to groundwater (Szaro 1989). Channelization can increase the intensity of extreme floods, because reductions in upstream storage capacity produce accelerated water flow downstream. Channelization also reduces the width of wooded riparian habitats, increasing the proportion of edge. Avian species richness has been shown to increase with the width of wooded riparian habitats (Stauffer and Best 1980).

Bank stabilization is typically used to protect property and structures from the impacts of flooding. Various manmade structures are used to protect banks and reduce the likelihood and impact of floods. Bank armor, such as rip-rap and levees, can protect stretches of bank and adjacent riparian vegetation, but can also lead to eddying and increased scouring of unprotected banks (DeBano and Heede 1987). In addition, bank armor reduces over-bank flooding, and consequently the occurrence of germination and regeneration of riparian vegetation. Under some conditions, certain types of flood-control structures can protect or enhance riparian habitat. For example, streamflow separations are used to create low energy flows at the bank. In so doing, separators can increase sediment deposition and create extensive stillwater areas adjacent to banks (DeBano and Heede 1987).

The riparian habitat that contains the largest known population of southwestern willow flycatchers along the Gila River in southwestern New Mexico is threatened by a combination of bank stabilization structures and agricultural practices within the floodplain (Phelps-Dodge Corporation 1995). Much of the floodplain is devoted to agricultural and ranching uses. Levees are used extensively along the border of agricultural fields to protect from flood damage. Riprapping, earthen dikes, and other structures are used along channel banks to further minimize flood damage. In some cases, the structures protect occupied flycatcher habitat. However, the combination of flood control structures in the channel, appropriation of the floodplain for agricultural or other uses, and the use of levees to further protect the land-uses occurring within the floodplain, has resulted in a system that isolates most of the floodplain, including existing flycatcher habitat, from natural flood processes needed to sustain and regenerate extensive new habitats.

Given that 25% of all known southwestern willow flycatchers breed at this site, the ramifications of potential habitat loss are substantial. Beyond these ramifications, however, this scenario points to a problem observed throughout the range of the southwestern willow flycatcher—development within floodplains. Be it homes, other types of structures, agricultural lands, or roads and bridges, development within floodplains increases the economic justification for flood control projects, which generally decreases opportunities for maintenance and restoration of floodplain processes necessary for the continual regeneration of riparian habitats (Poff et al. 1997).

Agricultural Development—The availability of irrigation water, relatively flat land and rich soils has spawned wide-scale agricultural development in river valleys throughout the Southwest. For example, more than 75% of the Mohave, Parker, Palo Verde, and Yuma valleys on the lower Colorado River has been converted to agriculture (Ohmart et al. 1986). These areas formerly contained vast riparian forests captured in early photographs of the area and probably comprised the most important riparian corridor in the Southwest. Collections of southwestern willow flycatcher nests made in the vicinity of Yuma in 1902 indicate that the flycatcher was at least locally very abundant along the lower Colorado River (Huels in litt. USFWS 1997b). The clearing of floodplain riparian habitat for agriculture continues today. For example, in January 1996, up to 2 km of occupied flycatcher habitat was lost to agricultural expansion on the Santa Ynez River in California (USFWS in litt.).

Livestock Grazing—Overgrazing by livestock has been a major factor in the modification and destruction of riparian habitats in the arid western U.S. (Fleischner 1996, Ohmart 1996, Dobkin et al. 1988). Riparian areas are often disproportionately preferred by cattle over surrounding uplands because of access to water, abundant and palatable forage, a cooler and shadier microclimate, and moderate slopes allowing easy access (Ames 1977, Glinski 1977, Szaro 1989; Fleischner 1996, Ohmart 1996). On uplands livestock act as geomorphic agents. By reducing vegetation cover and compacting soil, heavy livestock grazing reduces infiltration and increases runoff, erosion, and sediment yield, which can destabilize stream channels and affect the extent and distribution of riparian habitats (Trimble and Mendel 1995).

Grazing affects riparian vegetation through removal and trampling (Kauffman and Krueger 1984, Marlow and Pogacnik 1985). Removal by browsing affects the structure, spacing, and density of vegetation (Rea 1983, Cannon and Knopf 1984, Kauffman and Krueger 1984, Sedgwick and Knopf 1991). In several studies, willow canopy coverage was eight to ten times greater in areas excluded from grazing than in grazed areas (e.g., Taylor 1986, Schulz and Leininger 1990).

Grazing can also alter the age structure and species composition of riparian areas. Cattle readily eat shoots of cottonwood and willow, and heavy grazing can completely eliminate regeneration of these species (Glinski 1977, Rickard and Cushing 1982, Boles and Dick-Peddie 1983, Kauffman et al. 1983, Ohmart 1996). In contrast, cattle tend to avoid less palatable species such as saltcedar and juniper. Prolonged grazing in a riparian area can act as a selective agent shifting the relative abundance of plant species over time (Szaro and Pase 1983, Kerpez and Smith 1987). Dobkin et al. (1998) found that livestock grazing in riparian meadows resulted in a loss of perennial flow and a conversion of obligate wetland plant species and riparian bird species to upland species. When livestock were removed, perennial flow returned, as did obligate wetland plant species and an avian community comprised of wetland rather than upland species.

Trampling by livestock contributes to soil compaction, streambank erosion, widening and deepening of channels, increased runoff, and physical destruction of vegetation (Kauffman and Krueger 1984, Marlow and Pogacnik 1985, Szaro 1989, Trimble and Mendel 1995). In turn, unstable stream banks lead to accelerated erosion and increased sediment loads, which can destabilize floodplains and threaten the persistence of riparian habitats.

The impacts of grazing on riparian vegetation vary with the intensity and season of grazing. Late autumn and winter grazing may have relatively little effect, at least compared with other disturbances such as flooding (Kauffman and Krueger 1984, Knopf et al. 1988, Sedgwick and Knopf 1991). However, late spring and summer grazing typically has severe impacts, and results in little or no recruitment of riparian vegetation. This produces even-aged, non-reproducing communities of mature cottonwoods and decadent willows, with little understory. Such decadent, park-like stands, which are common throughout grazed drainages in the Southwest, are not suitable for southwestern willow flycatchers (Kauffman and Krueger 1984, Knopf et al. 1988, see Chapter 9).

In several studies, Willow Flycatcher numbers increased following the reduction or elimination of cattle grazing in riparian areas (Taylor 1986, Taylor and Littlefield 1986, Knopf et al. 1988). Harris et al. (1987) reported a 61% increase in flycatcher numbers over five years after grazing was reduced. Recent removal of livestock from the Riparian National Conservation Area on the upper San Pedro River in Cochise County, Arizona has resulted in both a dramatic increase in the recruitment of riparian vegetation and in the abundance of avian species reliant on dense understories, including the southwestern willow flycatcher, which was recently confirmed as a breeding species on the upper San Pedro (Kreuper 1993, McCarthey et al. 1998).

Low-intensity grazing during the non-growing season may be compatible in certain floodplain systems (i.e., those in proper functioning condition [USBLM 1993] and containing the full complement of riparian plant species and successional habitat types). For example, the Kern River Preserve in Kern County, California permits occasional, short duration and highly supervised livestock grazing in a small portion of the Preserve where meadows interface with riparian forest (R. Tollefson, pers. comm.). Livestock use of the Preserve, however, is not part of any annual grazing scheme. Furthermore, use is based on current ecological conditions, permitted at the discretion of and with the supervision of the Preserve Manager, and only permitted during the non-growing season. In the Gila Valley in southwestern New Mexico, livestock grazing occurs in irrigated pastures adjacent to the riparian stringers occupied by the largest known concentration of southwestern willow flycatchers (Parker and Hull 1994). In that case livestock forage is provided in the adjacent irrigated pasture and livestock use the riparian habitat primarily for shade. Neither of these cases represent a typical grazing situation for the Southwest, however. In the context of riparian management for the southwestern willow flycatcher, the appropriateness of a particular livestock grazing regime (in the uplands or riparian areas) should be evaluated based on current ecological conditions, the ecological potential for an area to support flycatcher habitat in the absence livestock grazing, and on the potential for livestock to serve as a magnet for cowbirds.

Although not yet documented for the southwestern willow flycatcher, livestock have been documented destroying - through trampling - willow flycatcher nests placed low in vegetation (Valentine et al. 1988). This should be considered a threat at any site within the southwestern willow flycatcher's range where flycatcher nest placement averages 3 m or less and where livestock are present during the breeding season.

Wild ungulates can also adversely impact riparian habitats, particularly when population densities are high. Elk (*Cervus canadensis*) have been shown to preclude the recovery of willow habitats even after the cessation of livestock grazing (Case and Kauffman 1997). Where elk and livestock are sympatric, reversing impacts to riparian areas may require more intensive management of both species. Elk occur in areas currently inhabited by southwestern willow flycatchers, including the higher elevation flycatcher sites in Arizona, New Mexico, Colorado, and Utah. The extent to which elk are adversely affecting areas inhabited by southwestern willow flycatchers is thought to be substantial in certain areas, however, quantitative studies that characterize the nature of impacts (e.g., extent, season, numbers of elk) are lacking.

Phreatophyte Control—In some areas riparian vegetation is still removed from waterways (streams and irrigation ditches) by mowing, cutting, rootplowing or spraying of herbicides. The intent of these practices is to increase watershed yield, remove impediments to stream flow, and limit water loss through evapotranspiration (Horton and Campbell 1974). As a consequence, riparian habitat is eliminated entirely or is maintained as a mosaic of very early successional patches not suitable for breeding flycatchers. Willow flycatcher populations (*E. t. adastus*) at the Malheur National Wildlife Refuge increased following the elimination of willow cutting and spraying (Taylor and Littlefield 1986).

Recreation—In the Southwest, campgrounds and recreational activities are concentrated in riparian areas because of accessibility, the presence of water, fishing opportunities, shade, and aesthetic qualities. These recreational activities include off-road vehicle use, boating, fishing, hunting, camping, birdwatching, hiking, swimming, floating, picnicking, and river rafting. The magnitude of such activities can be considerable. For example, Johnson and Carothers (1982) reported that the Glen Canyon and Lake Mead National Recreation Areas in Arizona received eight to nine million visitors per year. Recreation can impact riparian vegetation through damage or destruction of plants, elimination of seedlings, promoting invasion by exotic species, increased incidence of fires, indirect effects from soil compaction, and bank erosion (Johnson and Carothers 1982).

Disturbance from human recreation can reduce both the density and diversity of avian communities (Aitchison 1977, Szaro 1980, Taylor 1986, Riffell et al. 1996). In riparian areas in Utah, the presence of willow flycatchers was negatively correlated with campgrounds (Blakesley and Reese 1988). Food scraps and garbage in areas of high recreational use attract larger birds (e.g., jays, ravens) and small mammals (skunks, squirrels) which prey on bird nests and recently-fledged young (Johnson and Carothers 1982, Blakesley and Reese 1988). However, Haas (pers. comm.) reported a pair of southwestern willow flycatchers successfully fledging young from a nest that was several meters from a picnic table used frequently on weekends.

Urban Development—Urban development can result in a multitude of impacts to riparian habitats, such as the placement of homes and buildings within floodplains; the development of reservoirs and flood control structures within natural channels; overdraft of groundwater supplies and dewatering of streams and rivers; degradation of plant communities from heavy recreational use; increases in native and exotic predators; and improper placement of bridges. Some of these threats are discussed elsewhere in this chapter. Bowler (1990) documented the loss of riparian habitats in southern California that resulted from urban growth. One area of particular importance to the southwestern willow flycatcher and riparian habitats is the impacts of roads and bridges.

Southwestern willow flycatchers have been directly affected by roads and bridges that bisect riparian habitat. For example, an Empidonax flycatcher (probably a willow flycatcher) was killed by an automobile on a rural road that bisects willow flycatcher habitat in the White Mountains of Arizona (Sferra et al. 1995). In the San Juan Pueblo of New Mexico, placement of a new bridge across the Rio Grande resulted in the direct loss of habitat that contained two flycatcher territories (USFWS 1996). In Arizona, construction of a new bridge across the Gila River resulted in the loss of approximately one-third of a 1.5 ha riparian patch that supported four flycatcher territories (USFWS 1996). The number of territories decreased to one following habitat loss at that Graham County site (McCarthey 1998).

Placement of roads and bridges may have long-term effects of reducing overall habitat suitability for the willow flycatcher. Foppen and Reijnen (1994) and Reijnen and Foppen (1994) documented reduced breeding success, lower breeding densities, and higher dispersal rates of willow warblers (*Phylloscopus trochilus*) breeding next to roads that bisect forested habitat. Sogge (1995a) noted that the population decline and changes in the distribution of willow flycatcher territories on the Verde River in Arizona were consistent with other studies documenting adverse effects of roads that bisect habitat. However, Sogge (1995a) noted that the small size of that population coupled with sustained, high levels of predation and cowbird parasitism, may also have been factors at that site.

While the small size of sites and small number of territories involved in the above instances may not seem to justify conservation attention at first glance, it is important to keep in mind that these small instances of riparian habitat loss are numerous, frequent, and widespread (USFWS 1996, 1997b). At the minimum, these losses increase habitat fragmentation and reduce the carrying capacity of an area. Taken across the range of this species, the cumulative effects of these and other adverse impacts addressed in this chapter may result in destabilization of regional population dynamics.

Other Factors Contributing to Habitat Loss

Fire—Fire is a critical threat to occupied and unoccupied flycatcher habitat. In June of 1995, a fire on the Gila River in Pinal, County, Arizona, burned approximately six miles of riparian habitat potentially occupied by southwestern willow flycatchers (USFWS *in litt.*, USFWS 1997b). In 1996, five flycatcher breeding sites were degraded or lost altogether to fire, including two sites on the Rio Grande in New Mexico, one of the largest flycatcher sites on the San Pedro River in Arizona (Paxton et al. 1996), and two additional areas on the Gila River in Arizona where approximately eight miles of riparian habitat burned. In 1997 a fire started by an adjacent landowner burned a 32-ha portion of the Escalante Wildlife Area near Delta, Colorado (Owen and Sogge 1997). That location comprised one of the largest known breeding sites for willow flycatchers in Colorado with approximately seven pairs occupying the site in 1996.

Although fires are known to have occurred in riparian habitats historically, riparian habitats are not fire-adapted nor are they fire-generated communities. Thus, fires in riparian habitat are typically catastrophic. Busch (1995) documented that the current frequency and intensity of fires in riparian habitats is greater than what occurred historically because: (1) a greater accumulation of fuels due to a reduced frequency of scouring floods; and (2) the expansion and dominance in many areas of saltcedar (*Tamarix chinensis*), which is highly flammable. The increased incidence of fire is causing profound alterations in riparian habitats throughout the Southwest. Both saltcedar and arrowweed (*Tessaria sericea*) recover more rapidly from fire and are more tolerant of fire-induced increases in salinity and decreases in soil moisture than are cottonwood and willow (Busch and Smith 1993, Busch 1995). Consequently, saltcedar and arrowweed are becoming increasingly dominant in low elevation riparian habitats, and cottonwood and willow less so. On the lower Colorado River alone, Busch and Smith (1993) and Busch (1995) documented 166 individual fires that burned more than 11,800 ha between 1981 and 1990. Given the rate and extent of loss documented by Busch and Smith, and that the remaining cottonwood-willow habitat on the lower Colorado River is virtually surrounded by saltcedar, the potential for fire to result in further losses of the remaining cottonwood-willow habitat is substantial.

Exotic Species—The exotic tamarisk, or saltcedar, was introduced from Asia as an ornamental and erosion-control agent in the 1800s. It began spreading rapidly throughout the Southwest during the early part of the 20th Century (Tellman 1998). Today it has become dominant along many watercourses replacing multi-layered, multi-species native communities with monotypic stands uniform in structure. Hunter et al. (1987b) estimated that saltcedar dominated 49% of the area encompassed by riparian habitats in the Southwest, and occurred as a minor component in considerably more.

With its deep root system and extended production of seed from March through October, saltcedar thrives or persists where surface flow has been reduced or lost (Warren and Turner 1975, Horton 1977, Minckley and Brown 1982). The development of reservoirs and the concomitant change in flood regimes essential to the establishment of native riparian communities has enabled saltcedar to replace native broadleaf species. Furthermore, saltcedar establishment often results in a self-perpetuating regime of periodic fires. Fires were uncommon in native riparian communities prior to invasion by saltcedar, due to high moisture content in fuels and rapid removal of litter through decomposition and floods (Bradley et al. 1992). Consequently, native species are fire-intolerant. In contrast, saltcedar regenerates rapidly after fire (Busch and Smith 1993, Busch 1995). Areas with saltcedar that are not flooded regularly build up accumulations of salts in the soil, rendering the soil inhospitable for reestablishment of native species (Kerpez and Smith 1987).

Finally, the displacement of cottonwood-willow by saltcedar, particularly at elevations below 365 m, may reduce thermal buffering provided by the canopies of native riparian trees (C. Hunter pers. comm.). The absence or overall low reproductive success of mid-summer breeding birds at elevations below 365 m may be tied closely to a combination of (1) thermal tolerance of bird eggs being exceeded at ambient temperatures above 42° C (Walsberg and Voss-Roberts 1983); (2) predictable summer temperatures that frequently exceed 42° C during June and July (Hunter 1988, Hunter and Ohmart unpubl. manuscript); and (3) loss of most cottonwood-willow forests that may have provided effective thermal cover prior to the 1930s (e.g., Rosenberg et al. 1991). Hunter (pers. comm.) speculates that anticipated increases in average global temperature may exacerbate potential problems with productivity and distribution for mid- to late-summer breeding species such as the southwestern willow flycatcher.

In spite of the adverse impacts associated with the spread of saltcedar, this species is now a naturalized component of Southwestern drainages, particularly in Arizona, New Mexico, southern Utah, and southern Nevada. There is considerable irony in the fact that certain saltcedar habitats now provide what appears to be suitable nesting habitat for the endangered flycatcher (see Chapter 9)! That irony is reinforced by the fact that federal agencies responsible for recovering the southwestern willow flycatcher are also expending funds to "control" saltcedar. While saltcedar control may have some merit in systems for which the hydrological regime and water quality could truly support native riparian trees and shrubs, current control efforts and planning are focused almost exclusively on the symptoms rather than the root of the problem. Those involved in saltcedar management should heed Ewel's (1986) observation that "species invasions often reflect the conditions of the community being invaded rather than the uniquely aggressive traits of the invader." In the case of saltcedar, water management and water quality are the key factors. Control programs that do not consider these factors in the design of a restoration program run the risk of further reducing the biological diversity of an area, and, possibly, eliminating nesting habitat for the southwestern willow flycatcher. At a minimum, any area slated for saltcedar control or management should be thoroughly surveyed for flycatchers well in advance of physical alterations so that potential impacts to flycatchers can be fully evaluated and avoided.

Other exotic species have spread in riparian habitats throughout the range of the southwestern willow flycatcher. Russian-olive (*Elaeagnus angustifolia*) is abundant at middle elevations in New Mexico and Colorado (Szaro 1989). Where it occurs it is sometimes used for nesting by southwestern willow flycatchers (e.g., Skaggs 1996). Russian-olive appears to be less invasive than saltcedar and competitively inferior to native overstory species (Knopf and Olson 1984). Where found in mixed stands with native species, Russian-olive commonly occurs in less moist sites along the outer edge of riparian patches (Knopf and Olson 1984). Russian-olive supports a relatively high diversity and density of bird and mammal species, and may provide equivalent or better nesting habitat, although quantitative data are lacking (Knopf and Olson 1984). In California, giant reed (*Arundo donax*) is spreading rapidly. It forms dense monotypic stands unsuitable for flycatchers. Other exotic trees, such as Siberian elm (*Ulmus pumilis*) and tree of heaven (*Ailanthus simaruba*) occur in riparian areas within the flycatcher's range and do not appear to have any value as nesting substrates for flycatchers. At present their distribution is highly localized, which suggests that impacts to the flycatcher may be limited to local changes in riparian community composition.

Factors Directly Affecting Flycatchers

Cowbird Parasitism

Brood parasitism by brown-headed cowbirds (*Molothrus ater*) is a major threat to some populations of the southwestern willow flycatcher (Brown 1988, Harris 1991, Whitfield 1990, Sogge et al. 1997). The ecology of the cowbird is discussed in detail in Chapter 8. Originally thought to be commensal with American bison (*Bison bison*), cowbird numbers have increased tremendously with the expansion of livestock grazing, agriculture, and forest cutting (Laymon 1987, Robinson et al. 1993, Rothstein 1994). Cowbirds do not raise their own young, but rather lay their eggs in the nests of other species thus directly affecting their hosts by reducing nest success. Cowbird parasitism reduces host nest success in several ways. Cowbirds may remove some of the host's eggs, reducing overall fecundity. Hosts may abandon parasitized nests and attempt to renest, which can result in reduced clutch sizes, delayed fledgling, and reduced overall nesting success and fledgling survivorship (Whitfield 1994, Whitfield and Strong 1995). Cowbird eggs, which require a shorter incubation period than those of many passerine hosts, hatch earlier, giving cowbird nestlings a competitive advantage over the host's young for parental care (Bent 1960,McGeen 1972, Brittingham and Temple 1983).

Where studied, high rates of cowbird parasitism have coincided with southwestern willow flycatcher population declines (Whitfield 1994, Sogge 1995a, b, Whitfield and Strong 1995, Sogge et al. 1997), or, at a minimum, resulted in reduced or complete elimination of nesting success (Muiznieks et al. 1994, Whitfield 1994, Maynard 1995, Sferra et al. 1995, Whitfield and Strong 1995). Whitfield and Strong (1995) found that flycatcher nestlings fledged late in the season had a significantly lower rate of survival, and that cowbird parasitism was often the cause of delayed fledging.

A second brood parasitic species, the bronzed cowbird (*Molothrus aeneus*), is sympatric with *E. t. extimus* in portions of its range. However, except for one possible instance in the Gila River valley of New Mexico (Skaggs 1996) and one instance at Roosevelt Lake in Arizona (Sferra et al. 1995), the southwestern willow flycatcher is not known to be a host of this species (Lowther 1995). The bronzed cowbird is unlikely to pose any significant threat to *E. t. extimus* because it has a very limited distribution within the range of *E. t. extimus*, occurs at much lower densities than the brown-headed cowbird, prefers open habitats, and tends to prefer larger hosts, especially Icterids (Lowther 1995).

Predation

For many flycatcher populations, nest predation is the major cause of nest failure (Chapter 6). Most monitored populations experience high rates of nest predation ranging from 14 to 60% (Spencer et al. 1996, Whitfield and Strong 1995, Sferra et al. 1997, Sogge et al. 1997). Known or suspected nest predators include various snakes, predatory birds including corvids, owls, hawks, grackles and cowbirds, and small mammals including raccoons, ringtails, weasels, and rats (McCarthey et al. 1998).

Rates of predation may increase in human-altered landscapes. In the lower Colorado River valley, Rosenberg et al. (1991) noted increases in great-tailed grackles, a common nest predator. Increases in the extent of habitat fragmentation have been correlated with increased rates of nest predation in both forested and non-forested habitats (Picman et al. 1993, Askins 1993, Robinson et al. 1995). Whitfield (1990) noted that predation on flycatcher nests increased with decreasing distance to edge. Most small bird species in North America experience moderate rates of nest predation (30 to 60%) and the southwestern willow flycatcher, presumably, has adapted to similar rates. The key factor to determine is whether impacts, such as habitat fragmentation, are resulting in substantially higher rates of predation.

Parasites and Disease

Parasites and diseases can be critical factors affecting avian survival and reproduction, but tend to be poorly known (Dobson and May 1986). A variety of internal and external parasites have been recorded to affect willow flycatchers (Boland et al. 1989; Chapter 6 and references therein). However, the impact of such parasites or diseases on flycatchers is unknown.

Environmental Toxins

Where flycatcher populations are in proximity to agricultural areas, the use of pesticides poses a potential threat. Birds may be affected through direct toxicity or a reduction of their insect prey base. Although no quantitative data are available, physical deformities in willow flycatchers may indicate exposure to toxic compounds. Bill deformities and missing eyes have been reported from birds at sites in Arizona, Colorado and New Mexico (Paxton et al. 1997). In addition, flycatchers may be exposed to potentially toxic compounds on wintering or migration grounds.

In the lower Colorado River area, water management operations may exacerbate potential effects to flycatcher reproduction by concentrating naturally-occurring selenium. Selenium and other contaminants have been found in elevated levels in other birds within the lower Colorado River area (King and Andrews 1996). Selenium levels are known to be high at the Escalante State Wildlife Area in Colorado, where a willow flycatcher nestling was found with skull and bill deformities.

Summary

The above discussion illustrates the wide scope and magnitude of threats faced by this subspecies rangewide. The impacts documented during the last four years, alone, are alarming. Moreover, both small and large flycatcher populations have been adversely impacted or remain threatened. Haig et al. (1993) observed for the red-cockaded woodpecker, an endangered bird that still numbers from one thousand to several thousand pairs, that,

> "...species with such small populations are easily 'nickel and dimed' to extinction. That is, loss of a few small populations does not cause concern, but the cumulative effects of these losses could be dramatic. Therefore, a first step to species' recovery will be to stop these local extinctions."

The losses sustained by the flycatcher and current threats have been the subject of considerable conservation and research, public scrutiny, and litigation. However, we have yet to witness widescale application of what Haig et al. termed the "first step." This is evidenced by the numerous federal actions that have resulted in or are anticipated to result in the loss of flycatcher habitat and the displacement of flycatchers (USFWS 1997b). The cumulative effect of the threats and adverse impacts addressed in this chapter is substantial, and may account for the current low and relatively isolated population status for this subspecies. The rangewide scope and, in some cases, intense magnitude of these impacts underscores the critical need to protect existing flycatcher breeding groups and their habitat so as to not increase the degree of isolation among breeding groups. It also reinforces the concept of habitat conservation and management at the scale of the drainage (see Chapter 3), with the goal of decreasing habitat isolation and providing for population movement that results from population phenomena (i.e., emigration, dispersal), stochastic events (e.g., catastrophic floods, fires), or deterministic events (e.g., inundation of habitat).

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